# Differential Controls of Greenhouse Gas (CO2, CH4, and N2O) Concentrations in Natural and Constructed Agricultural Waterbodies on the Northern Great Plains

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**Abstract** Inland waters are hotspots of greenhouse gas (GHG) cycling, with small water bodies particularly active in the production and consumption of carbon dioxide ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$). However, wetland ponds are being replaced rapidly by small constructed reservoirs in agricultural regions, yet it is unclear whether these two water body types exhibit similar physical, chemical, and environmental controls of GHG content and fluxes. Here, we compared the content and regulatory mechanisms of all three major GHGs in 20 pairs of natural wetland ponds and constructed reservoirs in Canada's largest agricultural region. Carbon dioxide content was associated primarily with metabolic indicators in both water body types; however, primary production was paramount in reservoirs, and heterotrophic metabolism a stronger correlate in wetland ponds. Methane concentrations were correlated positively with eutrophication of the reservoirs alone, while competitive inhibition by sulfur-reducing bacteria may have limited $CH_4$ in both waterbody types. Contrary to expectations, $N_2O$ was undersaturated in both water body types, with wetlands being a significantly stronger and more widespread $N_2O$ sink. Varying regulatory processes are attributed to differences in age, depth, morphology, and water-column circulation between water body types. These results suggest that natural and constructed water bodies should be modeled separately in regional GHG budgets.

**Plain Language Summary** Small inland water bodies, ponds and reservoirs, are hotspots of greenhouse gas (GHG) production. However, modern farming practices replace natural wetlands by artificial reservoirs, without consideration on the effects on landscape GHG fluxes. Here we measured concentrations and controls of carbon dioxide ($CO_2$), methane ($CH_4$), and nitrous oxide ($N_2O$) in locally-paired wetland ponds and agricultural reservoirs and found that while GHG concentrations were similar in the two waterbody types, underlying mechanisms differed, owing to the differences in shape, depth, and age of individual water bodies. Shallower wetland ponds were typically more fully mixed, exhibited more oxygen at depth, and had higher nutrients and organic carbon concentrations reflecting their ages. Findings suggest that the balance between wetland drainage and reservoir construction may have a pronounced effect on regional budgets of GHG fluxes, particularly in the face of forthcoming climate changes.

## 1. Introduction

Small inland waterbodies are well-known hotspots of greenhouse gas (GHG) emissions (Cole et al., 2007; Tranvik et al., 2009) owing to their cumulative abundance in many regions (Downing et al., 2006) and because they often release carbon dioxide ($CO_2$) and methane ($CH_4$) at higher rates than larger inland waters (Downing, 2010). However, GHG concentrations are extremely variable, both spatially and temporally in these small surface waters (Deemer et al., 2016; Downing, 2010; Holgerson & Raymond, 2016; Jensen et al., 2022), with some sites acting as net GHG sinks (Webb, Hayes, et al., 2019; Webb, Leavitt, et al., 2019). Measurements of all three major GHGs ($CO_2$, $CH_4$, and nitrous oxide ($N_2O$)) are providing insights into spatial and temporal patterns of GHG flux (Huttunen et al., 2002; Jensen et al., 2022; Liikanen et al., 2006; Whitfield et al., 2011), but it is not currently known whether different water body types (natural vs. artificially constructed) respond similarly to underlying mechanisms controlling gas fluxes. These uncertainties lead to questions of the accuracy of regional upscaling models and estimates of the importance of small surface waters in calculations of net global warming potential

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of inland waters (Webb, Hayes, et al., 2019; Webb, Leavitt, et al., 2019). Given the global increase in constructed reservoirs (Clifford & Heffernan, 2018; Webb, Leavitt, et al., 2019), combined with the extensive and continued drainage of natural wetland ponds in agricultural landscapes (Cortus et al., 2011), it is crucial to be able to differentiate the magnitude and controls of GHG flux between the two water body types.

Greenhouse gas emissions from wetland ponds have been investigated because of their high global abundance, large sediment carbon pools, and high rates of $CH_4$ emissions (Bridgham et al., 2012; Holgerson & Raymond, 2016; Tian et al., 2016). In the northern Great Plains, there are millions of small prairie wetlands within the "Prairie Pothole Region" that spans 715,500 $km^2$ across three Canadian provinces (Alberta, Saskatchewan, Manitoba) and five US states (Montana, North Dakota, South Dakota, Minnesota, Iowa) (Bortolotti et al., 2016; Euliss et al., 1999). Such prairie wetlands have been suggested as net carbon sinks suitable for GHG mitigation (Badiou et al., 2011), due to their high productivity and organic carbon sequestration in sediments (Baulch et al., 2021).

Drainage associated with agricultural expansion has eliminated >40% of wetland ponds in the northern Great Plains (Cortus et al., 2011). Restoration of these wetlands often takes one of two forms; refilling of drained lowland deposits or construction of new agricultural reservoirs. In the first case, wetlands have been re-established in select areas during the past two decades by building berms and allowing the wetland to refill with precipitation and runoff. Resultant water bodies often exhibit higher rates of $CO_2$ and $CH_4$ emission than do undisturbed wetland sites, at least during the first decade after restoration (Bortolotti et al., 2016; Euliss et al., 1999). In contrast, de novo construction of artificial reservoirs is often sited where they will be most useful for agricultural needs and often exhibit different morphology (steep sides, greater depths ~4m), compared to natural wetland ponds (<2 m depth). While previous work has provided insights into spatial and temporal controls of net GHG flux from small constructed reservoirs (Jensen et al., 2022; Webb, Hayes, et al., 2019; Webb, Leavitt, et al., 2019), it is currently unclear how regulatory mechanisms of farm reservoirs differ from those of natural wetland ponds.

Greenhouse gas dynamics can differ between larger natural (e.g., lakes) and human-constructed (e.g., reservoirs) waterbodies owing to fundamental differences in their morphology, productivity, and watershed characteristics (Hayes et al., 2017). Large reservoirs used to generate hydroelectric power or provide drinking water can experience extreme water level fluctuations, leading to changes in hydrostatic pressure and periodic $CH_4$ ebullition when the water level is reduced (Galy-Lacaux et al., 1999; Joyce & Jewell, 2003; Keller & Stallard, 1994). These reservoirs also typically have very high catchment area to surface area ratios, leading to increased influx of nutrients and organic matter, higher productivity, and elevated respiration and methanogenesis (Davidson et al., 2015; West et al., 2016). In contrast, smaller reservoirs, particularly those in agricultural lands, are used for watering of livestock and to provide relief from both flooding and drought. Such small agricultural reservoirs usually exhibit lower catchment:surface area ratios and their water levels, while variable, are rarely actively managed. Known regionally as "dugouts", small reservoirs often receive substantial nutrient runoff due to their position in the landscape, with adjacent land being used mainly for crop or pastureland. Given these observations, differences in GHG fluxes between natural ponds and small constructed reservoirs are unlikely to follow patterns seen when comparing larger lakes and reservoirs. Given that wetland ponds and agricultural reservoirs will experience comparable weather and catchment loading, we anticipate that any differences in their GHG fluxes will be attributable to variation in water body shape (depth, surface area: volume ratios) and age (affecting organic matter quantity and quality).

Agricultural intensification is increasing the prevalence of small artificial waterbodies across landscapes (Clifford & Heffernan, 2018; Webb, Leavitt, et al., 2019). In contrast to natural wetland ponds, agricultural reservoirs are younger, relatively deeper, and exhibit less emergent vegetation. Worldwide, there are ~440,000 reservoirs ≤0.1 $km^2$ covering >75,000 $km^2$ of agricultural land (Downing et al., 2006). Of the limited research to date, studies from Australia (Grinham, Albert, et al., 2018; Ollivier et al., 2018, 2019), and India (Panneer Selvam et al., 2014) show that small agricultural reservoirs typically emit $CO_2$, $CH_4$, and $N_2O$ to the atmosphere. In contrast, work on the northern Great Plains in Canada suggests that while all reservoirs are sources of $CH_4$, more than half of sites act as sinks for $CO_2$ (52%) and $N_2O$ (67%) (Webb, Hayes, et al., 2019; Webb, Leavitt, et al., 2019), with predictable seasonal patterns of gas flux (Jensen et al., 2022), suggesting high variation among global agricultural regions.

To accurately upscale GHG emissions from small waterbodies to larger regions, we need a more comprehensive understanding of the differences in mechanisms regulating production of $CO_2$, $CH_4$, and $N_2O$ in wetland ponds and agricultural reservoirs. In this study, we present a comparative analysis of $CO_2$, $CH_4$, and $N_2O$ concentrations,
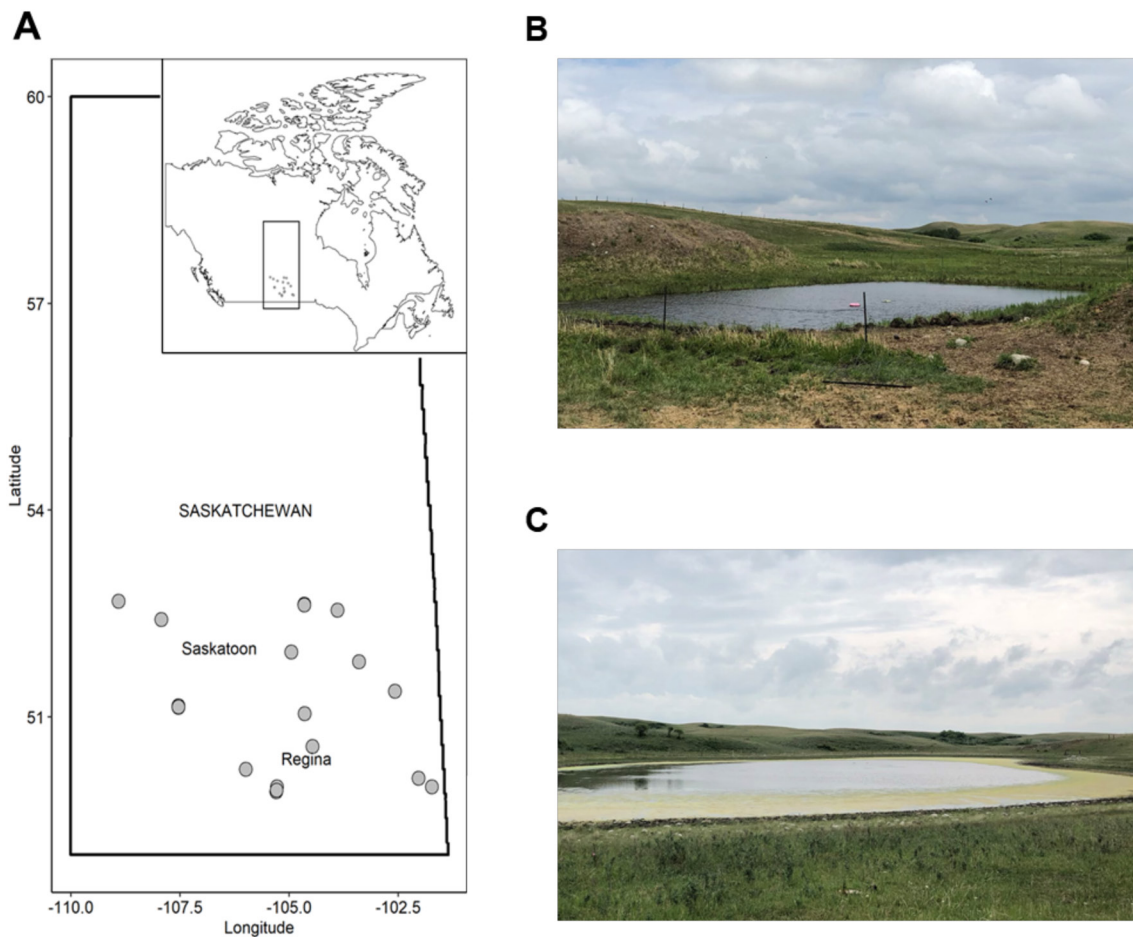
**Figure 1.** (a) Map of locations sampled. Gray circles indicate each of the 20 paired sites sampled during June and July of 2019. Each site is composed of one agricultural reservoir and one natural wetland pond. (b) Photo of a typical agricultural reservoir. (c) Photo of a natural wetland pond. Photo credit: Sydney Jensen.

estimated diffusive flux, and regulatory factors between natural prairie wetlands and small agricultural reservoirs in the Northern Great Plains, the largest agricultural region in Canada. The aim of this study was to identify whether GHG concentrations were of similar magnitude in natural wetlands and agricultural reservoirs, and evaluate site types differences in physical, chemical and environmental controls of GHG levels in each type of waterbody. To achieve these goals, GHG concentrations were measured in 20 natural wetland ponds paired with 20 co-located agricultural reservoirs in southern Saskatchewan, Canada. By comparing GHG dynamics in paired sites, we were able to isolate the unique effects of basin characteristics (age, morphology, hydrology), since landscape factors such as nutrient loading, climate, and land-use were comparable with individual locales. We used generalized additive models (GAMs) to determine the best explanatory covariates of GHG fluxes in each type of waterbody. The results of this study will inform the need for small constructed reservoirs to be considered separately from natural wetland ponds in global upscaling models.

## 2. Materials and Methods

We sampled 20 agricultural reservoirs and 20 natural wetland ponds located within the grassland region of Saskatchewan, Canada, for this study (Figure 1a). Artificially constructed reservoirs have steep banks and deep basins, used for reliable water storage for livestock watering, while wetland ponds are created by natural processes and usually have shallow depths. Both waterbody types receive water from surface, subsurface, or groundwater flow, and are typically closed basins with no outlets and study sites were hydrologically isolated during our study period, but can "fill and spill" during extremely wet periods (Coles & McDonnell, 2018). Each reservoir was paired with a nearby natural wetland to form a single sampling site (mean separation = 280 m, min = 81 m,

max = 763 m), with similar landuse around both waterbodies, and no intervening human-made structures (roads, buildings). Each site was sampled once, within a one-month period between 19 June and 15 July 2019.

## 2.1. Field Collection

When possible, sampling took place from a canoe at the deepest location of each waterbody, although very shallow natural wetlands were sampled by wading, with efforts made to minimize sediment perturbation. Samples were collected between 8:45 and 16:30 hr. A Yellow Springs Instrument (YSI) multi-parameter probe was used to measure water temperature (°C), dissolved oxygen (DO, mg L$^{-1}$, % saturation), salinity (g total dissolved solids [TDS] L$^{-1}$), conductivity (μS cm$^{-1}$), and pH at ∼0.5-m depth intervals from the surface to the bottom. Parameters were estimated at 0.25-m intervals in very shallow wetlands. DO sensors were calibrated prior to measurement at each sampling location. Maximum depth of each waterbody was estimated using a Norcross Hawkeye hand-held depth finder. Average wind speed was measured using a Kestral handheld wind meter at ∼1 m above the water surface. Water samples were collected using a submersible pump at ∼0.5 m depth into a pre-rinsed carboy, filtered through 80-μm mesh to remove large zooplankton and particulate matter, and transported to the laboratory within 4 hr of collection.

Concentrations of GHG were estimated using the in-field headspace extraction method (Jensen et al., 2022; Webb, Hayes, et al., 2019). Briefly, water was collected from ∼0.5 m depth using a submersible pump to overfill a 1.2-L glass serum bottle. The bottle was then sealed with a two-way rubber stopper, 60-mL of atmospheric air was added to the bottle as 60-mL of water was simultaneously removed, and was then shaken vigorously for 2 min to equilibrate the air and water. Two replicate gas samples were extracted from the headspace using an air-tight syringe and put into 12-mL pre-evacuated Exetainer vials with double wadded caps. This process was repeated to achieve a total of four gas samples per waterbody, while an atmospheric sample was also collected and placed in a similar vial. All gas samples were stored at room temperature until analysis, within 2 months of collection.

## 2.2. Laboratory Analysis

Water samples for chemical analyses were filtered within 24 hr of collection and stored at 4°C. Duplicate water samples for chlorophyll *a* (Chl *a*) were filtered through a Whatman GF/C filter (1.2 μm nominal pore size), wrapped in foil, and frozen (−10°C) until analysis. Filtrate was then passed through a sterile 0.45-μm pore polycarbonate filter, transferred to a 1-L amber glass bottle with no headspace, and stored in the dark at 4°C until analysis at the Institute for Environmental Change and Society, University of Regina. Nutrient concentrations (soluble reactive phosphorus (SRP), total dissolved phosphorus (TDP), total dissolved nitrogen (TDN), nitrate and nitrite (NO$_x$), ammonia (NH$_3$)) were measured using standard analytic procedures on a Lachat QuikChem 8500 (Swarbrick et al., 2022) and carbon concentrations (dissolved organic carbon (DOC), and dissolved inorganic carbon (DIC)) were measured on a Shimadzu model 5000A total carbon analyzer (Swarbrick et al., 2022). Chlorophyll *a* concentration was measured using standard trichromatic spectrophotometric methods (Finlay et al., 2009; Jeffrey & Humphrey, 1975).

CO$_2$, CH$_4$, and N$_2$O samples were analyzed at the Global Institute for Water Security, University of Saskatchewan as detailed in Jensen et al. (2022). Briefly, headspace gas samples were analyzed for the dry molar fraction of each gas using a fully calibrated Scion 456 Gas Chromatograph (Bruker Ltd.) with Combipal autosampler (CTC Analytics—PAL System), using argon as the carrier gas. N$_2$O was measured using a micro-electron capture detector and argon/methane (90/10) as a makeup gas (injector temperature 60°C, column temperature 60°C, detector temperature 350°C). All gases were calibrated using mixed gas standards (Praxair/Linde with balance gas as N$_2$) to determine the relationship between peak area and concentration. In addition to the mixed gas standards, one N$_2$O-only standard (0.1 ppmv single Scotty) was added for N$_2$O calibrations to help improve accuracy of the non-linear peak area-concentration relationship shown for N$_2$O. For N$_2$O, six calibration standards were available; however, typically four standards were used with concentrations ranging from 0.100 ppmv to 0.98 ppmv, reflective of the standard data range. The linear peak area-concentration relationships (CO$_2$ and CH$_4$) or curve fit (N$_2$O) was further tested via manual dilutions of certified standards. Concentrations of CO$_2$ and CH$_4$ were consistently well above detection limits, with N$_2$O the only gas ever nearing detection limits. For our analyses, the minimum measured N$_2$O concentration in a sample vial was 0.151 ppmv. As noted the lowest N$_2$O standard was 0.1ppmv, and method detection limit was 0.025 ppmv.

### 2.3. Numerical Analyses

The dry molar fractions of $CO_2$, $CH_4$, and $N_2O$ were corrected for dilution with on-site atmospheric air and converted to concentrations according to the gas-specific solubility coefficients, atmospheric pressure, and salinity (Weiss, 1974; Weiss & Price, 1980; Yamamoto et al., 1976). The mean of four replicate gas was used as estimates of $CO_2$, $CH_4$, and $N_2O$ at each site. Dissolved concentrations of each gas were used to estimate the diffusive flux ($f_C$) of $CO_2$, $CH_4$, and $N_2O$ using the gas transfer velocity ($k_c$), gas concentration of the water ($C_{water}$), and the ambient air concentration ($C_{air}$) following equation (Equation 1):

$$f_C = k_c(C_{water} - C_{air}). \tag{1}$$

Mean air concentrations were 413.09 µatm for $CO_2$, 1.868 µatm for $CH_4$, and 0.333 µatm for $N_2O$. Diffusive flux was calculated using the average gas transfer velocity as measured directly on comparable agricultural reservoirs in 2017 ($k_{600}$-$CO_2$ = 1.50 m d$^{-1}$; $k_{600}$-$CH_4$ = 1.64 m d$^{-1}$, Webb, Hayes, et al., 2019; $k_{600}$-$N_2O$ = 1.64 m d$^{-1}$) following procedures of Webb, Leavitt, et al. (2019) and Jensen et al. (2022).

Water-column stratification was estimated by calculating the maximum squared Brunt-Väisälä buoyancy frequency (s$^{-2}$). We identified the steepest in situ density gradient observed for water temperature profiles taken at 0.5-m intervals (0.25 m in shallow waterbodies) and s$^{-2}$ was calculated using the *rLakeAnalyzer* package (Read et al., 2012) in R (version 4.0.5; R Core Team, 2021) following Webb, Leavitt, et al. (2019).

To determine if GHG concentrations were significantly different between natural and constructed waterbodies, we used a one-sample Wilcoxon signed rank test with $\alpha = 0.05$. The Wilcoxon test was chosen over a one-sample *t*-test because pond data were not normally distributed.

Carbon dioxide equivalent ($CO_2$-eq) flux was calculated using the 100-year sustained flux global warming potential model if fluxes were above zero, and with the cooling potential model if fluxes were negative (Neubauer & Megonigal, 2015). Diffusive flux values of $CH_4$, and $N_2O$, were converted to the same units before multiplication by the appropriate warming or cooling potential, and summation across gases to give the total $CO_2$-eq flux for each site (Neubauer & Megonigal, 2015).

To quantitatively evaluate the similar and dissimilar underlying mechanistic controls of GHG in wetland ponds and agricultural reservoirs, we used generalized additive models (GAMs, Wood, 2017; Wood et al., 2016) following Webb, Hayes et al. (2019), Webb, Leavitt et al. (2019), and Jensen et al. (2022). Covariates included as predictors in the models were chosen based on the basis of prior knowledge of biotic and abiotic factors known to influence production or consumption of each GHG (Jensen et al., 2022; Webb, Hayes, et al., 2019; Webb, Leavitt, et al., 2019). Variables used as smoothed terms included nutrients (DIN, and SRP), organic carbon (DOC), DOC to nutrient ratios (DOC: NO$_x$, DOC:SRP), productivity (Chl *a*, and DO), and stratification strength (buoyancy frequency). We included conductivity in the $CH_4$ model as a proxy for sulfate concentrations, the main regional anion. All modeling was done using the *mgcv* software package (Wood, 2011; Wood et al., 2016) within the R computational environment (version 4.0.5; R Core Team, 2021).

Waterbody type was included as a parametric fixed effect in the models to evaluate GHG characteristics of wetland ponds differed from those of constructed reservoirs, and a factor-smooth interaction was used for each smooth term to allow the smooth effects of each covariate in the model to vary between ponds and reservoirs. The gas concentration ($CO_2$ and $CH_4$) was assumed to be conditionally distributed gamma (loglink function) resulting in a GAM that is suitable for positive continuous responses. No link function was used for $N_2O$ as it was normally distributed without transformation. To ensure the assumptions of the models were not violated, we evaluated basis size, dispersion of residuals, homogeneity of variance, and the relationship between the observed and predicted responses. Selection of smoothness parameters was evaluated using residual marginal likelihood (REML) (Wood, 2011). To help with model selection, the double penalty approach of Marra and Wood (2011) was used. Parameters predicting $CO_2$, $CH_4$, and $N_2O$ concentrations in wetland ponds and constructed reservoirs were considered significant at 95% confidence level ($\alpha = 0.05$).

## 3. Results

### 3.1. Water Quality

Physical and chemical conditions varied between agricultural reservoirs and natural wetland ponds (Table 1). In general, wetland ponds were shallower (0.6 m) than reservoirs (2.7 m) and had higher concentrations of DOC,

**Table 1**
*Physical and Chemical Parameters for Pairs of 20 Agricultural Reservoirs and 20 Natural Wetland Ponds During 2019*

| Parameter | Units | Mean (SD) Reservoirs | Mean (SD) Wetlands |
|---|---|---|---|
| Water Temperature | °C | 20.4 (2.8) | 20.6 (3.1) |
| Max Depth | Meters | 2.7 (1.0)*** | 0.6 (0.5) |
| Relative Depth | % | 8.3 (4.2)*** | 1.1 (1.5) |
| Buoyancy Frequency | $s^{-2}$ | 0.011 (0.01)*** | 0.005 (0.009) |
| Wind Speed | km h$^{-1}$ | 7.05 (4.10) | 9.64 (5.54) |
| pH | Unitless | 8.8 (0.7) | 9.1 (0.6) |
| Surface DO | % Sat. | 95.7 (32.8) | 109.5 (38.7) |
| Deep DO | % Sat. | 24.5 (39.2) | 84.8 (48.6)*** |
| Conductivity | µS cm$^{-1}$ | 1495.4 (1460.2) | 3,336.9 (5078.0) |
| DIC | mg C L$^{-1}$ | 51.5 (25.8) | 60.5 (34.6) |
| DOC | mg C L$^{-1}$ | 28.9 (12.9) | 53.8 (29.2)*** |
| Alkalinity | mg L$^{-1}$ | 226.1 (108.9) | 295.7 (155.5) |
| TDP | µg P L$^{-1}$ | 248.5 (385.3) | 449.5 (600.3)** |
| SRP | µg P L$^{-1}$ | 204.8 (344.6) | 341.0 (568.4) |
| TDN | µg N L$^{-1}$ | 2048.8 (1107.4) | 3,515.5 (2087.6)*** |
| NO$_x$ | µg N L$^{-1}$ | 272.3 (367.6) | 113.0 (129.1) |
| NH$_3$ | µg N L$^{-1}$ | 71.1 (103.3) | 100.8 (91.4) |
| Chl $a$ | µg L$^{-1}$ | 43.7 (85.8) | 13.2 (16.7) |
| CO$_2$ | µM | 30.1 (44.3) | 37.6 (90.4) |
| CH$_4$ | µM | 1.3 (1.7) | 1.4 (1.7) |
| N$_2$O | nM | 8.3 (2.2)*** | 6.3 (1.6) |

*Note.* Values presented as mean ± standard deviation. Significance represents results from a Wilcoxon rank tests at *$p < 0.1$; **$p < 0.05$; ***$p < 0.01$. Relative depth is $Z_{max}$ as a percentage of the mean diameter. Surface DO = dissolved oxygen ~10 cm below the surface; Deep DO = dissolved oxygen ~10 cm above the sediment. DIC = dissolved inorganic carbon. DOC = dissolved organic carbon. TDP = total dissolved phosphorus. SRP = soluble reactive phosphorus. TDP = total dissolved nitrogen. NO$_x$ = nitrate and nitrite. NH$_3$ = ammonia. Chl $a$ = chlorophyll $a$. CO$_2$ = carbon dioxide. CH$_4$ = methane. N$_2$O = nitrous oxide, max = maximum.

DIC, TN, TP, and DO concentrations. In contrast, reservoirs had significantly greater relative depths (ratio of max. depth to average diameter, 8.3%) compared to wetlands (1.1%). Concentrations of SRP, NOx, and ammonia were comparable between wetlands and reservoirs, as were surface water temperatures and Chl $a$ concentrations, although reservoirs had a much larger range of Chl $a$ than did natural wetlands (Table 1). Conductivity was not statistically different between the waterbody types, although wetlands' mean was skewed by two high values (14,899 and 18,975 µS cm$^{-1}$).

### 3.2. Greenhouse Gas Concentrations

Concentrations of CO$_2$ and CH$_4$ were comparable between natural wetland ponds and agricultural reservoirs, while N$_2$O was significantly lower in wetland environments (Table 1). Eight reservoirs and 14 wetlands were undersaturated with respect to CO$_2$ whereas all wetlands and 18 reservoirs were supersaturated with CH$_4$. Most reservoirs and wetlands were undersaturated in N$_2$O, with only one wetland and four reservoirs being supersaturated.

### 3.3. Controls of Greenhouse Gas Concentration

A GAM including only pH showed a very strong relationship between pH and CO$_2$ concentrations in both reservoirs and wetlands, explaining 83% of deviance and exhibiting a strong negative relationship between pH and CO$_2$ concentrations in both waterbodies (Figure 2). When pH was not included in the GAMs, CO$_2$ concentrations were best estimated by a combination of physical and chemical parameters in both agricultural reservoirs and natural wetland ponds explaining 87.7% of deviance (Figure 3). However, in this case, predictors of CO$_2$ concentrations differed between waterbody types with significant predictors ($p < 0.05$) in reservoirs including DO saturation (negative), chlorophyll $a$ (positive), and ratios of DOC:NO$_x$ (negative), whereas in wetlands parameters included alkalinity (positive), buoyancy frequency (positive) and DOC:NO$_x$ (unimodal).

Methane concentrations were estimated best in a GAM using a combination of physical, chemical, and biological predictors for both waterbodies together, with 86.2% deviance explained (Figure 4). Significant predictors ($p < 0.05$) of CH$_4$ in reservoirs included surface DO saturation (negative), chlorophyll $a$ (positive), conductivity (unimodal), buoyancy frequency (negative), and DOC:NOx (positive). In wetlands, the significant predictors of CH$_4$ were conductivity (negative) and buoyancy frequency (positive).

Nitrous oxide concentrations were estimated best in a GAM using a combination of biotic and abiotic parameters, resulting in 95.8% of deviance explained (Figure 5). Predictors were similar between reservoirs and wetlands, with surface DO saturation, NO$_x$ concentration, and chlorophyll $a$ being significant, although the nature of the relationships differed between waterbodies. Wetlands exhibited decreasing N$_2$O concentrations as DO increased, while reservoirs exhibited a positive relationship with DO. Reservoirs showed increasing N$_2$O concentrations with increasing NO$_x$ concentrations, whereas wetlands exhibited decreasing gas concentrations. Both waterbodies had significant relationships between N$_2$O concentrations and Chl a concentrations, with a negative relationship in reservoirs, and a unimodal relationship in wetlands with maximum N$_2$O at ~18 µg N L$^{-1}$. Additionally, DOC concentration was weakly positive in reservoirs, whereas wetlands exhibited a decreasing trend in N$_2$O as DOC increased.

### 3.4. Greenhouse Gas Fluxes

Mean CO$_2$ diffusive flux calculated over all waterbodies was 16.5 mmol m$^{-2}$ day$^{-1}$ (SD = 59.4), while CO$_2$ flux values were slightly less in reservoirs (13.0 ± 37.5 mmol m$^{-2}$ day$^{-1}$) than in wetlands (20.1 ± 76.2 mmol m$^{-2}$ day$^{-1}$;
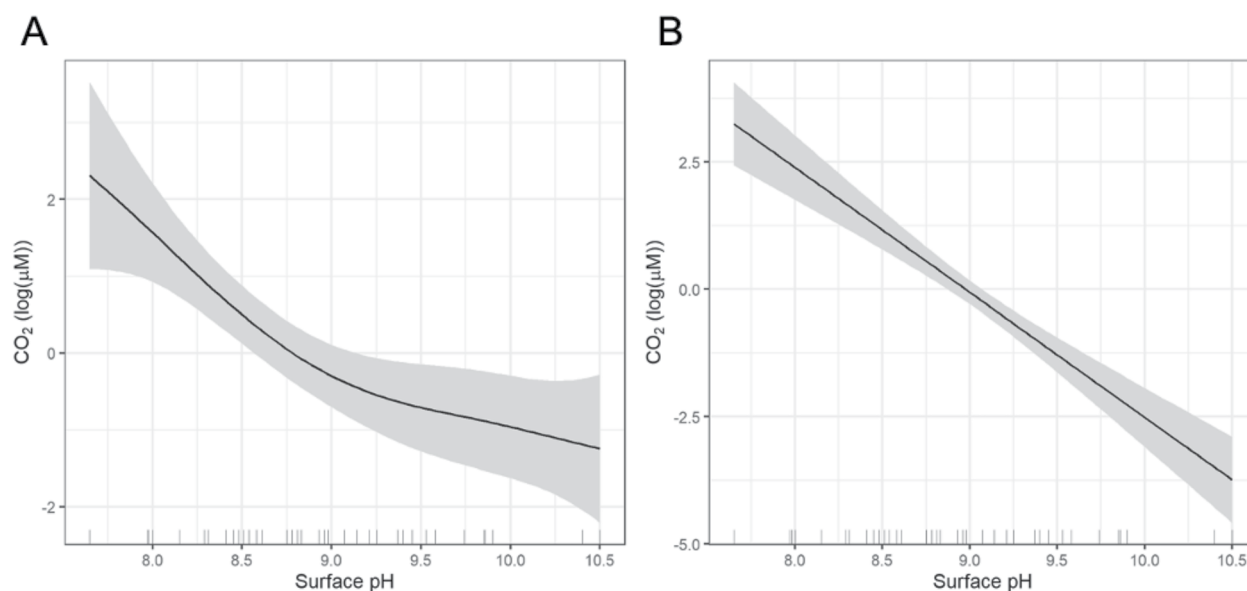
**Figure 2.** Partial effects plots from the generalized additive model assessing the relationship between pH and $CO_2$ concentration in (a) reservoirs and (b) wetlands. Deviance explained = 83%. Gray shaded region represents the 95% Bayesian credible interval. Rugs (vertical ticks) on *x*-axis represent measured data points.

Figure 6a). Mean $CH_4$ diffusive flux over all sites was $1.2 \pm 1.4$ mmol m$^{-2}$ day$^{-1}$, with similar values in reservoirs ($1.1 \pm 1.4$ mmol m$^{-2}$ day$^{-1}$) and wetlands ($1.2 \pm 1.9$ mmol m$^{-2}$ day$^{-1}$; Figure 6c). Finally, mean diffusive flux of $N_2O$ flux in all waterbodies was $-1.2 \pm 1.8$ μmol m$^{-2}$ d$^{-1}$, while that of wetland ponds was $-2.1 \pm 1.3$ μmol m$^{-2}$ d$^{-1}$ and reservoirs was $-0.4 \pm 1.9$ μmol m$^{-2}$ d$^{-1}$, with 20% of the sites being net sources of $N_2O$ to the atmosphere (Figure 6b).

## 4. Discussion

Loss of natural wetlands, combined with increased construction of small farm reservoirs, has the potential to alter landscape patterns of GHG flux from continental grasslands. Here we found that while fluxes of $CO_2$ and $CH_4$ did not differ substantially between constructed reservoirs and wetland ponds, presumptive regulatory mechanisms varied among waterbody types suggesting potential differences in response to future environmental change (Figures 3–5). The variation likely reflected differences in basin morphology and stratification (Kalff & Downing, 2016; Qin et al., 2020), as well as DOC quality or quantity (Waiser & Robarts, 2004). Furthermore, while previous work has demonstrated undersaturation of $N_2O$ in agricultural reservoirs (Jensen et al., 2022; Webb, Leavitt, et al., 2019), this study found that natural wetlands were more substantial sinks of atmospheric $N_2O$ than artificial waters. Differences in regulatory mechanisms suggest that wetland ponds and agricultural reservoirs need to be considered separately in inland water emission models and development of future GHG management strategies for farms.

### 4.1. Physical and Chemical Comparison of Wetland Ponds and Agricultural Reservoirs

While agricultural reservoirs and natural ponds had comparable concentrations of most measured water chemistry variables, waterbody morphometry and thermal stratification were significantly different between surface water types (Table 1), with potential cascading effects on concentrations of DOC, TDN, and TDP. Elevated DOC, TDN, and TDP levels may be attributable to older age, shallower depth, and a less stable water column in the wetland ponds (Kalff & Downing, 2016). Similarly, a review of 573 global lakes showed that TN and TP concentrations decrease with increasing water depth, in part reflecting differences in mixing depth and the area of sediment in contact with the overlying water (Qin et al., 2020). Agricultural reservoirs were often thermally stratified during the monitoring period (data not shown), while natural wetlands were not due to their shallow depth (mean = 0.6 m), larger surface area (mean = 16,425 m$^2$), and potential greater exposure to wind due to absence of lateral excavation spoil piles (Kalff & Downing, 2016; Holgerson et al., 2022, but this was not observed in our data set, Table 1). We observed a strong correlation between wetland depth and SRP (Spearman $p = -0.61$; Figure S2 in Supporting Information S1); however, only weak to moderately-strong relationships ($p = -0.04$ and $-0.31$) between
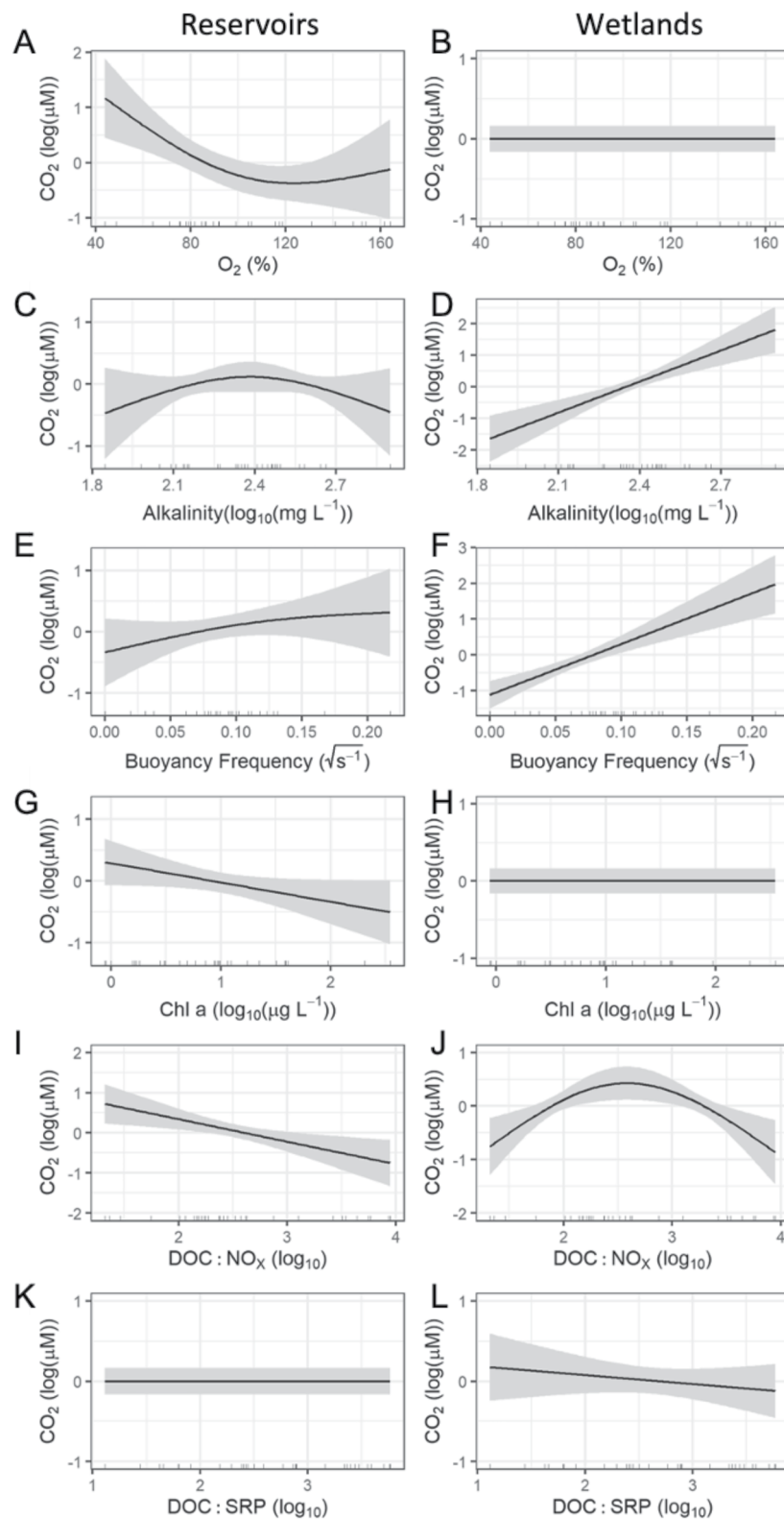
**Figure 3.** Partial effects plots from the generalized additive model relating $CO_2$ in reservoirs (left) and wetlands (right) to water column conditions. Overall deviance explained = 87.7%. Gray shaded region represents the 95% Bayesian credible interval. Rugs along the *x*-axis represent the measured data points. Covariates included dissolved oxygen (a, b), alkalinity (c, d), buoyancy frequency (e, f), chlorophyll *a* (Chl *a*; G, H), ratios of dissolved organic carbon to nitrate/nitrite (DOC:$NO_3^-$; i, j), and ratios DOC to soluble reactive phosphorus ratio (DOC:SRP; k, l).

**Figure 4.** Partial effects plots from the generalized additive model relating dissolved $CH_4$ concentration to environmental conditions in reservoirs (left) and wetlands (right). Deviance explained = 86.2%. Gray shaded region represents the 95% Bayesian credible interval. Rugs along the *x*-axis represent the measured data points. Covariates include surface dissolved oxygen ($O_2$) (a, b), deep $O_2$ (c, d), chlorophyll *a* (Chl *a*; E, F), conductivity (g, h), buoyancy frequency (i, j), ratios of dissolved organic carbon (DOC) to NOx (DOC:$NO_x$; k, l) and ratios of nitrate/nitrite to soluble reactive phosphorus ($NO_x$:SRP; m, n).
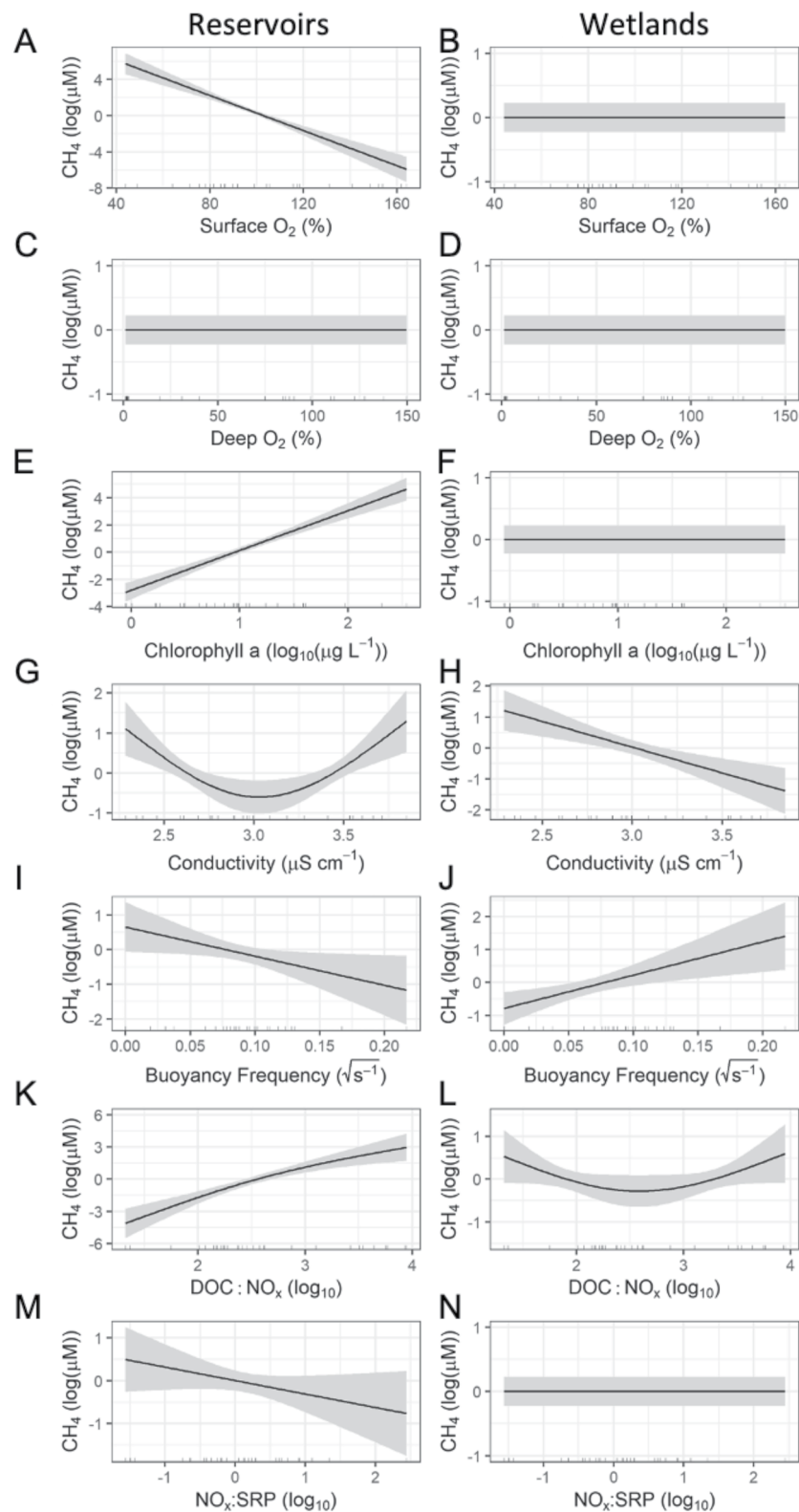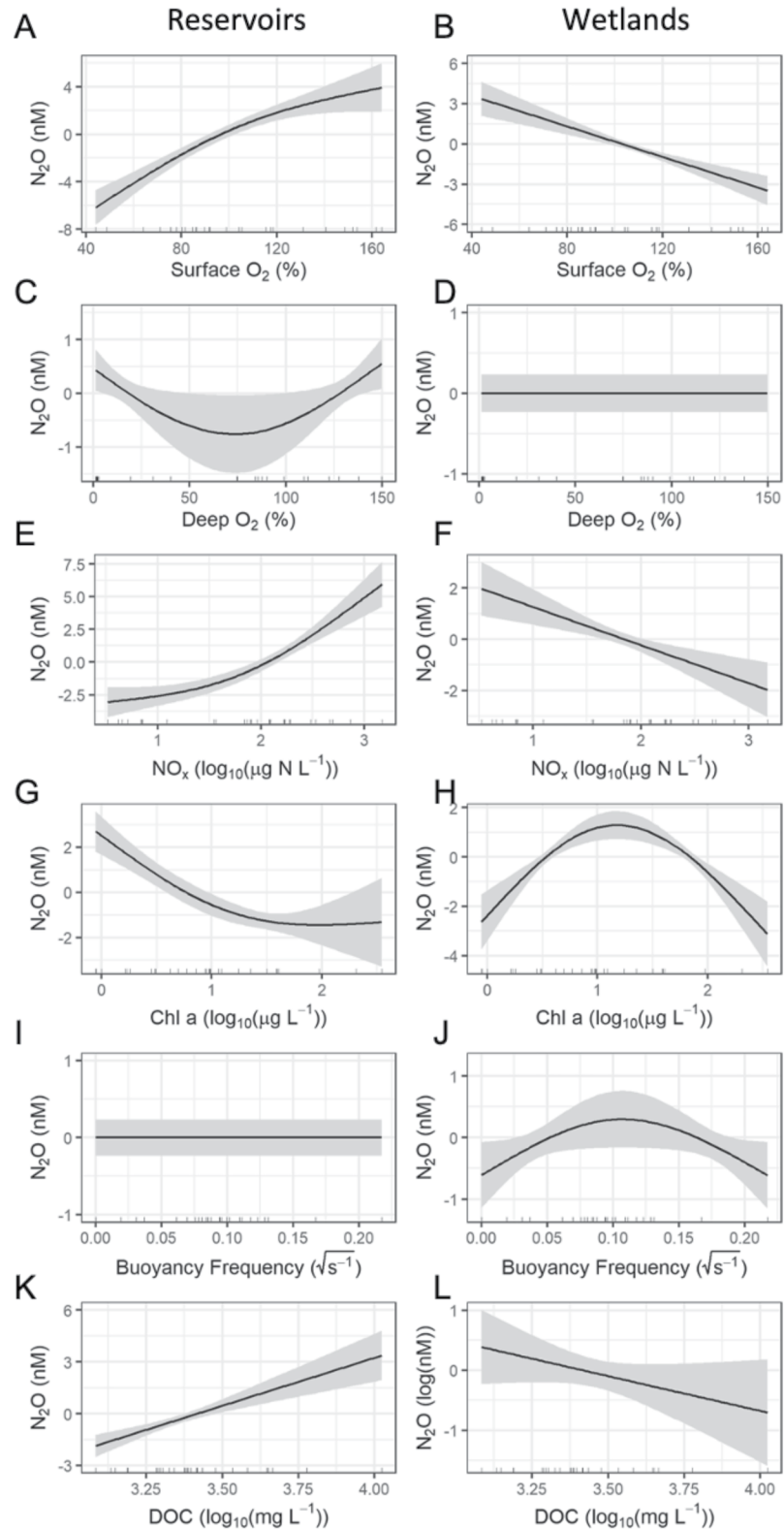
**Figure 5.** Partial effects plots from the generalized additive model relating nitrous oxide ($N_2O$) concentrations to environmental conditions in reservoirs (left) and wetlands (ight). Deviance explained = 95.8%. Gray shaded region represents the 95% Bayesian credible interval. Rugs along the *x*-axis represent the measured data points. Covariates include surface dissolved oxygen ($O_2$; (a, b), deep $O_2$ content (c, d), nitrate/nitrite ($NO_x$; (e, f), chlorophyll *a* (Chl *a*; G, H), buoyancy frequency (i, j), and dissolved organic carbon (DOC; k, l).

**Figure 6.** Diffusive fluxes of (a) $CO_2$ (mmol m$^2$ day$^{-1}$), (b) $N_2O$ (umol m$^2$ day$^{-1}$), and (c) $CH_4$ flux (mmol m$^2$ day$^{-1}$). (d) $CO_2$-equivalent (eq) flux aggregated by each gas. All panels presented as mean $\pm$ standard deviation. Note difference in *y*-axis scale among panels. Panel (d) error bars are calculated on the mean total $CO_2$-eq flux.
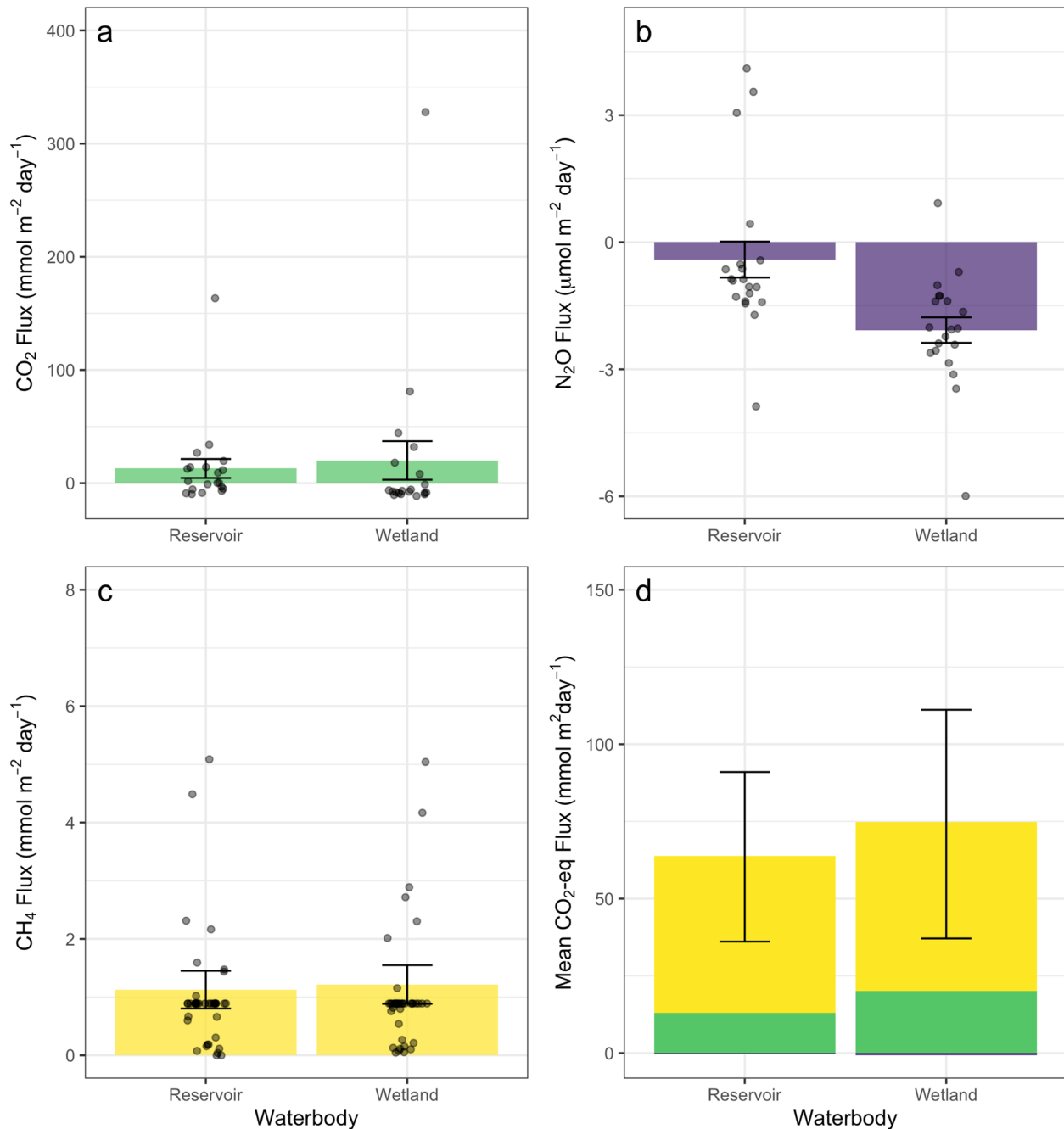
SRP, TDN and depth in reservoirs, while DOC was only moderately correlated to depth or stratification strength (as buoyancy frequency) in both waterbodies (Figures S1 and S2 in Supporting Information S1). These patterns suggest that differences in morphology arising from mode of origin (lowland flooding, construction) had substantial effects on chemical characteristics that impacted GHG production and loss from the different waterbody types.

### 4.2. Regulation of Carbon Dioxide Concentration

Carbon dioxide concentrations were comparable in reservoirs and natural ponds, with gas levels correlated strongly with changes in pH, a pattern characteristic of hardwater lakes with pronounced carbonate buffering

(Duarte et al., 2008; Finlay et al., 2015; Jensen et al., 2022; Stumm & Morgan, 1970; Webb, Leavitt, et al., 2019; Wiik et al., 2018). When effects of pH were removed from the models, $CO_2$ concentrations were found to be strongly influenced by metabolic processes in both waterbody types, with wetlands demonstrating evidence of primarily benthic respiration (DOC was a significant predictor, Chl $a$ was not; Figure 3), while reservoirs demonstrated a stronger influence of epilimnetic primary production (DO and Chl a were significant predictors; Figure 3).

In natural ponds, there was a significant unimodal relationship between $CO_2$ and DOC:$NO_x$ concentrations, but the lack of a relationship between $CO_2$ and Chl $a$ suggests that heterotrophic metabolism was controlling $CO_2$ concentrations (Bogard et al., 2017; Webb, Leavitt, et al., 2019) by remineralizing organic matter (Maher et al., 2019), likely in the sediments. While we observed a general relationship between buoyancy frequency and $CO_2$ concentrations in natural ponds, only seven of the 20 sites exhibited thermal stratification, suggesting that stratification was not a broadly important mechanism regulating $CO_2$. We had expected that wetland ponds would experience less profound or persistent thermal stratification than the deeper agricultural reservoirs, particularly given the larger surface areas and shallower depths characteristic of wetland ponds (Holgerson et al., 2022; Kalff & Downing, 2016). Interestingly, we did not observe any relationship between changes in $O_2$ and $CO_2$ concentrations in wetland ponds, in contrast to patterns seen in other shallow small ponds (Bortolotti et al., 2016; Holgerson, 2015) and our small agricultural reservoirs (Figure 3). We suggest that uncoupling of $CO_2$ and DO may reflect strong carbonate buffering, such as seen in other DIC-rich systems (Stets et al., 2017; Vachon et al., 2020), a hypothesis supported by the significant positive relationship between $CO_2$ concentrations and DIC (as alkalinity) (Spearman $p_{\text{DIC-alkalinity}} = 0.89$).

Primary production appears to have played a significant role in regulating $CO_2$ levels in agricultural reservoirs, as DO levels were correlated negatively ($P < 0.05$) with $CO_2$ concentrations, as seen in other microbially-regulated systems (Cole & Caraco, 2001; Hanson et al., 2006; Stets et al., 2017; Vachon et al., 2020). Similarly, the strong negative relationship observed between $CO_2$ and Chl $a$ is consistent with the role of autotrophs in regulating aquatic gas content (Figure 5), as previously seen in other agricultural reservoirs within this region (Webb, Leavitt, et al., 2019). This contrasts our previous findings that primary production was not a significant driver of seasonal $CO_2$ concentrations in constructed reservoirs (Jensen et al., 2022), and may reflect difference in the relative control of autotrophic and heterotrophic control on temporal (Jensen et al., 2022) and spatial scales (this paper). The coupling of near-surface $CO_2$ in reservoirs with DOC:$NO_x$ ratios, suggests that higher autotrophic activity depletes $NO_x$ content in this N-limited region (Bergbusch et al., 2021; Glibert et al., 2016; Swarbrick et al., 2019; Waiser et al., 2011).

### 4.3. Regulation of Methane Concentrations

Analysis with GAMs revealed that $CH_4$ concentrations were correlated negatively to DO levels in reservoirs, suggesting the presence of anoxic $CH_4$ production (Figure 4; Megonigal et al., 2004, see also Bogard et al., 2014). In addition, we observed the lowest $CH_4$ concentrations in reservoirs where $O_2$ was supersaturated in surface waters. However, the lack of any relationship between $CH_4$ and deep water DO suggests that surface $CH_4$ was more strongly impacted by consumption in the water column rather than production at depth (D'Ambrosio & Harrison, 2021; Hanson & Hanson, 1996, Holgerson, 2015). As such, surface water $CH_4$ concentration in the reservoirs appears to be regulated by $O_2$ changes resulting from primary productivity. In contrast, $CH_4$ levels in wetland ponds were unrelated to DO content, suggesting insufficient time for gas oxidation before release to the atmosphere in these substantially shallower systems (Holgerson 2015).

Stratification strength (as buoyancy frequency) was significant in GAMs for both reservoirs and wetlands, with a negative relationship with $CH_4$ in reservoirs and positive relationship in wetlands (Figure 4). The negative relationship in reservoirs is consistent with $CH_4$ remaining trapped in the cool deep water, as previously seen elsewhere (Bastviken et al., 2004; Holgerson, 2015; Juutinen et al., 2009; Kankaala et al., 2013), possibly allowing for more oxidation in the water column (Saarela et al., 2020). In contrast, the positive relationship between stratification strength and $CH_4$ levels seen in wetland ponds was unexpected, and may reflect a disproportionate effect of the seven thermally-stratified wetland ponds. Here, it may be that the very shallow wetland systems never stratify, and could simply exhibit more rapid equilibration with the atmosphere.

A negative relationship between $CH_4$ and surface water conductivity in wetland ponds (Figure 4) suggests that sulfate-reducing bacteria may have outcompeted methanogenic bacteria for metabolic substrates. Surfacewater

conductivity is highly correlated with sulfate concentrations (adj. $R^2 = 0.83$) in these systems and previous analyses (Jensen et al., 2022), suggesting that sulfate concentrations (up to 3,883 mg L$^{-1}$ in reservoirs; 6,687 mg L$^{-1}$ in ponds) can be high enough to inhibit methanogenesis in freshwaters (Lovley & Klug, 1983). Similar negative relationships between $CH_4$ concentrations and conductivity are known from other reservoirs in our study region (Webb, Leavitt, et al., 2019), as well as natural wetlands in North America and Europe (Gauci et al., 2004). Conductivity was also a significant predictor of $CH_4$ in the reservoirs; however, the relationship was unimodal with the lowest concentrations of $CH_4$ observed at conductivity values of approximately 1,000 μS cm$^{-1}$. This pattern contrasts our previous seasonal evaluations which identified a strong negative relationship between conductivity and $CH_4$ concentrations (Jensen et al., 2022). We hypothesize that, on a seasonal scale, increasing conductivity due to evapoconcentration during late summer combines with increased mixing as the water column becomes shallower, reduces $CH_4$ emissions from wetlands in August and Sept. Alternately, in deeper reservoirs, it may be that inhibition of methanogenesis is not as strong a determinant of $CH_4$ production as DO and stratification strength. Further research will be needed to resolve this issue.

The positive relationships between DOC:NO$_x$, Chl $a$, and $CH_4$ concentrations in agricultural reservoirs, but not wetland ponds, suggests that $CH_4$ production may be limited by the supply of labile organic matter supply in constructed waterbodies. Alternately, this difference between reservoirs and ponds may be attributed to the age of the system—agricultural reservoirs are typically only a few decades old, and may not have as high as supply of DOC (Table 1) to form labile substrates for methanogens, while wetland ponds have large pools of DOC that are often refractory (Waiser & Robarts, 2004). While sediments were not sampled from either waterbody, $CH_4$ levels are known to increase with sediment C content and C:N ratios in other regional agricultural reservoirs (Jensen et al., 2022; Webb, Leavitt, et al., 2019), supporting this possible mechanism.

### 4.4. Regulation of Nitrous Oxide Concentrations

On average, $N_2O$ concentrations were undersaturated in both agricultural reservoirs and natural ponds; however, wetlands were typically stronger $N_2O$ sinks than were constructed waterbodies, in part reflecting outgassing from four reservoirs (Figure 6; Table 1). Nitrous oxide concentrations were within the range seen in many lentic waterbodies (Table 2), with values below those characteristic of flowing waters (Webb et al., 2021). Specifically, prairie waterbodies sampled here had similar $N_2O$ concentrations to those seen in other regional agricultural reservoirs (Jensen et al., 2022; Webb, Hayes, et al., 2019), small polymictic lakes in Ireland (Whitfield et al., 2011), small boreal lakes and reservoirs of varying trophic status in Finland (Huttunen et al., 2002, 2003), and boreal ponds in Canada (Soued et al., 2016). However, ours is the first study to report such a high proportion (~95%) of undersaturation among wetland ponds.

Despite similar in situ values, $N_2O$ concentrations appeared to be regulated by different processes in small prairie ponds and constructed waterbodies (Figure 5). In principle, production of $N_2O$ arises from incomplete denitrification (Firestone & Davidson, 1989), nitrification (Baulch, Dillon, et al., 2011), dissimilatory nitrate reduction to ammonium (DNRA; Scott et al., 2008), and possibly other less well established processes (Domeignoz-Horta et al., 2015), whereas complete denitrification is the only process known to consume $N_2O$ (Quick et al., 2019). Each of these processes is affected by ecosystem productivity (Kemp & Dodds, 2002), the presence of oxygen in specific habitats (e.g., shallow and deepwaters, sediments; Knowles, 1982), substrate concentrations (e.g., inorganic nitrogen; Kemp & Dodds, 2002; Taylor & Townsend, 2010), and their effects on consequent microbial activities (Taylor & Townsend, 2010). Given that only denitrification is known to be capable of reducing $N_2O$ levels (Quick et al., 2019), and that most prairie waters were undersaturated with $N_2O$ (Figure 6), we infer that differences in regulatory processes between constructed wetlands and natural ponds (Figure 6) reflect variation in the locale or intensity of denitrification, with more intense consumption of $N_2O$ in natural wetland ecosystems.

In reservoirs, concentrations of $N_2O$ exhibited positive relationships with the degree of DO saturation, NO$_x$ concentration, and DOC content, whereas $N_2O$ levels declined with primary production, as Chl $a$ (Figure 5). Overall, these patterns are consistent with the predominant control operating via $N_2O$ consumption through denitrification (Baulch, Dillon, et al., 2011; Zhang et al., 2021), while the negative relationship with Chl $a$ suggests competition between primary producers and $N_2O$-producing microbes for nitrogen species in agricultural reservoirs (Jensen et al., 2022; Webb, Hayes, et al., 2019). For example, elevated production in deep or stratified systems would be expected to lead to deepwater anoxia that favors complete denitrification and consumption of extant $N_2O$. The positive relationships between $N_2O$ and NO$_x$ may reflect NO$_x$-limited denitrification in

**Table 2**
*Flux Values of Carbon Dioxide ($CO_2$), Methane ($CH_4$), and Nitrous Oxide ($N_2O$) Recorded Herein Along With Literature Values*

| Citation | System | Region | Method | $CO_2$ diffusive mmol $m^{-2}$ $day^{-1}$ | $CH_4$ diffusive mmol $m^{-2}$ $day^{-1}$ | $N_2O$ diffusive μmol $m^{-2}$ $day^{-1}$ | $CH_4$ ebullition mmol $m^{-2}$ $day^{-1}$ |
|---|---|---|---|---|---|---|---|
| Ollivier et al. (2018) | Agricultural Reservoirs | Australia | FC | 18.56 to 30.24 | 3.99 to 10 | - | - |
| Ollivier et al. (2019) | Agricultural Reservoirs | Australia | FC, BT | 13.21* | 0.29* | 3.05* | 8.8** |
| **Present Study** | **Agricultural Reservoirs** | **Saskatchewan, Canada** | HE | **−30 to 464** | **0 to 13.5** | **−9.66 to 11.61** | |
| Webb, Hayes, et al. (2019) | Agricultural Reservoirs | Saskatchewan, Canada | HE | - | - | −12 to 166 | - |
| Webb, Leavitt, et al. (2019) | Agricultural Reservoirs | Saskatchewan, Canada | HE | −21 to 466 | 0.14 to 92 | - | - |
| Peacock et al. (2021) | Artificial Waterbodies | Sweden | FC | 82.70* | 2.57* | - | 16.02* |
| D'Acunha and Johnson (2019) | Constructed Stormwater Wetlands | British Columbia, Canada | HE | 10.41 to 1,266 | 0 to 55.5 | −7.44 to 57.47 | - |
| McClure et al. (2020) | Eutrophic Reservoir | North America | FC, BT | - | 0.03 to 1.37 | - | 0.023 to 2.24 |
| Jeffrey et al. (2019) | Freshwater Subtropical Wetland | Australia | FC, BT | - | 1.5 to 66.2 | - | 5.5* |
| Bastviken et al. (2004) | Lakes | North America | HE, FC | - | 0.068 to 0.43 | - | 0.016 to 2.79 |
| DelSontro et al. (2016) | Lakes | Quebec, Canada | FC, BT | 52* | 2.2* | - | 1.1* |
| Strayer and Tiedje (1978) | Lakes | North America | HE, BT | - | 10 to 46 | - | 21* |
| Bortolotti et al. (2016) | Natural and Restored Wetland Ponds | Saskatchewan, Canada | HE | −60 to 1,350 | 0.7 to 13.3 | - | - |
| **Present Study** | **Natural Wetland Ponds** | **Saskatchewan, Canada** | HE | **−31 to 784** | **0.13 to 14.11** | **−13.5 to 2.2** | |
| DelSontro et al. (2016) | Ponds | Quebec, Canada | FC, BT | 57* | 4.2* | - | 4.6* |
| Huttunen et al. (2002) | Ponds | Finland | FC, BT | −0.03 to 2.50 | 0.11 to 4.55 | −2.95 to 3.86 | - |
| Badiou et al. (2011) | Prairie Wetland Ponds | Prairie Pothole Region, Canada | FC | - | 0.11 to 3.81 | 1.43 to 20.0 | - |
| McNicol et al. (2017) | Restored Wetland | California, USA | HE, FC | 95 to 3,773 | 0.035 to 7.42 | −8.64 to 103.68 | 1.15 to 2.3 |
| Natchimuthu et al. (2014) | Shallow Ponds | Sweden | FC | −9.8 to 16.0 | 3.3 to 13.1 | - | 3.1 to 11.6 |
| Grinham, Dunbabin, and Albert (2018) | Small Artificial Waterbodies | Australia | FC | - | <3.12 | - | >3.12 |
| Whitfield et al. (2011) | Small Headwater Lakes | Ireland | HE | −6.60 to 46 | 0 to 4.5 | −3.8 to 6.9 | - |
| Repo et al. (2007) | Small Wetland Lakes | Siberia | HE, FC | −1.59 to 84.07 | 0.069 to 7.48 | - | 0.062 to 0.686 |
| Yang et al. (2015) | Subtropical Aquaculture Ponds | China | FC | −79.1 to 50.9 | 2.15 to 141.9 | −0.57 to 17.72 | - |
| Villa et al. (2021) | Temperate Freshwater Marsh | Ohio, USA | FC | - | 0.78* | - | 0 to 95.04 |
| Prėskienis et al. (2021) | Tundra Lakes and Ponds | Nunavut, Canada | HE, BT | −9.25 to 23.39 | 0.108 to 0.955 | - | 0.04 to 5.255 |
| van Bergen et al. (2019) | Urban Pond | Netherlands | FC, BT | 79.22* | 1.66* | - | 8.31* |
| Panneer Selvam et al. (2014) | Various Inland Waters | India | FC | −28.2 to 253.2 | 0.9* | - | 7.0* |

*Note.* Diffusive flux values reported for $CO_2$, $CH_4$, $N_2O$, while ebullitive flux only reported for $CH_4$. Method used for diffusive flux: floating chamber (FC) or headspace extraction (HE); method used for ebullitive flux: floating chamber (FC) or bubble trap (BT). * = mean value, ** = modeled value. Bold values are highlighting the data from the study reported here.

the region (e.g., Gooding & Baulch, 2017). Specifically, higher $N_2O$ production via denitrification may occur where increased NOx concentrations stimulate this process, or increase the $N_2O$ yield of this process (Baulch, Schiff, et al., 2011). Denitrification is also potentially stimulated by higher DOC (Gooding & Baulch, 2017; Knowles, 1982), again consistent with relationships seen in reservoirs. Nitrification in reservoirs could also lead to the observed positive relationships between $N_2O$ and NOx via concurrent production of NOx and $N_2O$. Nitrification is highly oxygen-sensitive, and may be stimulated by increased $O_2$ availability (e.g., Kemp & Dodds, 2002; Rysgaard et al., 1994).

In contrast to results in reservoirs, $N_2O$ concentrations in natural wetland ponds decreased as the DO saturation in surface waters increased, $NO_x$ concentrations increased, and DOC content increased (Figure 5). Ponds, with their shallower mean depth, may have a greater importance of benthic processes and also have high habitat heterogeneity, which may yield complex relationships, consistent with multiple drivers affecting N cycling and $N_2O$ production.

The negative relationship between $N_2O$ and DO in wetlands, and the high degree of $N_2O$ undersaturation in well oxygenated waters is novel, as previous work has demonstrated $N_2O$ undersaturation under low oxygen conditions (Zhang et al., 2021). In shallow productive environments such as our study ponds, these patterns may emerge due to diffusion between anoxic sediments and overlying habitats which are often highly productive. However, while anaerobic denitrification is traditionally the only known biological pathway of $N_2O$ reduction, aerobic bacteria and cyanobacteria have been found to have the NosZ gene and are therefore capable of $N_2O$ consumption (Farías et al., 2013; Park et al., 2017; Rees et al., 2021). Other studies have also found evidence of assimilative $N_2O$ reduction into particulate organic nitrogen by marine cyanobacterial cultures (Farías et al., 2013) and biological $N_2O$ consumption has been observed in oxygenated marine waters (Cornejo et al., 2015; Rees et al., 2021). To our knowledge, such strong $N_2O$ undersaturation has not been observed before in highly oxygenated fresh waterbodies, hence it is also possible there is an alternative $N_2O$ consumption mechanism that may be decoupled from $NO_3^-$ reduction. Further experimentation will be required to evaluate the potential of this mechanism, as well as the relative importance of water column and sedimentary processes in controlling $N_2O$ fluxes in small prairie waterbodies.

### 4.5. CO2-Equivalent Fluxes

We found that wetland ponds were more frequently sinks for GHGs than were agricultural reservoirs (Figure 7), with agricultural reservoirs contributing comparable $CO_2$-eq to the atmosphere as natural wetlands (Figure 6d). Estimates of $CO_2$-eq flux in this study showed that ~5% of agricultural reservoirs and ~15% of wetland ponds are acting as net $CO_2$-eq sinks when calculated using the 100-year sustained-flux global warming and cooling potentials (Neubauer & Megonigal, 2015) (Figure 7). These proportions are similar to those reported for small agricultural reservoirs in Australia (Table 2), which found that the importance of individual GHG varied among $CO_2$ (55% of effect) $CH_4$ (42%) and $N_2O$ (3%) (Ollivier et al., 2019). $CO_2$-eq fluxes observed in these waterbodies were comparable to other natural (Bortolotti et al., 2016; Kankaala et al., 2013; Whitfield et al., 2011) and constructed waterbodies (Ollivier et al., 2018, 2019; Webb, Leavitt, et al., 2019), even though most previous studies do not include $N_2O$ fluxes (Table 2). Because $CH_4$ ebullition can account for up to 90% of total $CH_4$ emissions (Grinham, Albert, et al., 2018), but was not included in our study, we suggest that $CO_2$-eq fluxes are likely underestimated based on diffusive fluxes alone.

Our estimates of $CO_2$-eq flux assume constant diel flux rates and comparable $k$ values between reservoirs and wetland ponds. Previous research has observed considerable variability in $CO_2$, $CH_4$, and $N_2O$ fluxes over 24-hr periods in other regions. For example, $CH_4$ can exhibit midday maxima in concentration (Kim et al., 2015; Xing et al., 2004), while $CO_2$ levels often peak at night (Raymond et al., 2013; Wiik et al., 2018), and $N_2O$ has been observed to increase (Rosamond et al., 2011; Wu et al., 2018), decrease (Molina et al., 2021), and exhibit no change overnight (Baulch et al., 2012). These patterns are further complicated by potential diel variability in wind speed overnight (MacIntyre et al., 2021; Rudberg et al., 2021). Here, we chose to calculate $CO_2$-eq flux using observed daytime values and extrapolating over the full 24-hr period, because an earlier investigation of these regional ponds did not reveal any evidence of consistent diel variability of GHG fluxes (Jensen et al., 2022). We hypothesize that the high alkalinity of our study systems (mean > 200 mg $L^{-1}$), and elevated sulfate in groundwater and subsurface water, may have dampened diel variability in $CO_2$ and $CH_4$ flux, respectively (Jensen et al., 2022), but recognize that more systematic and extensive sampling is required in order to confirm this
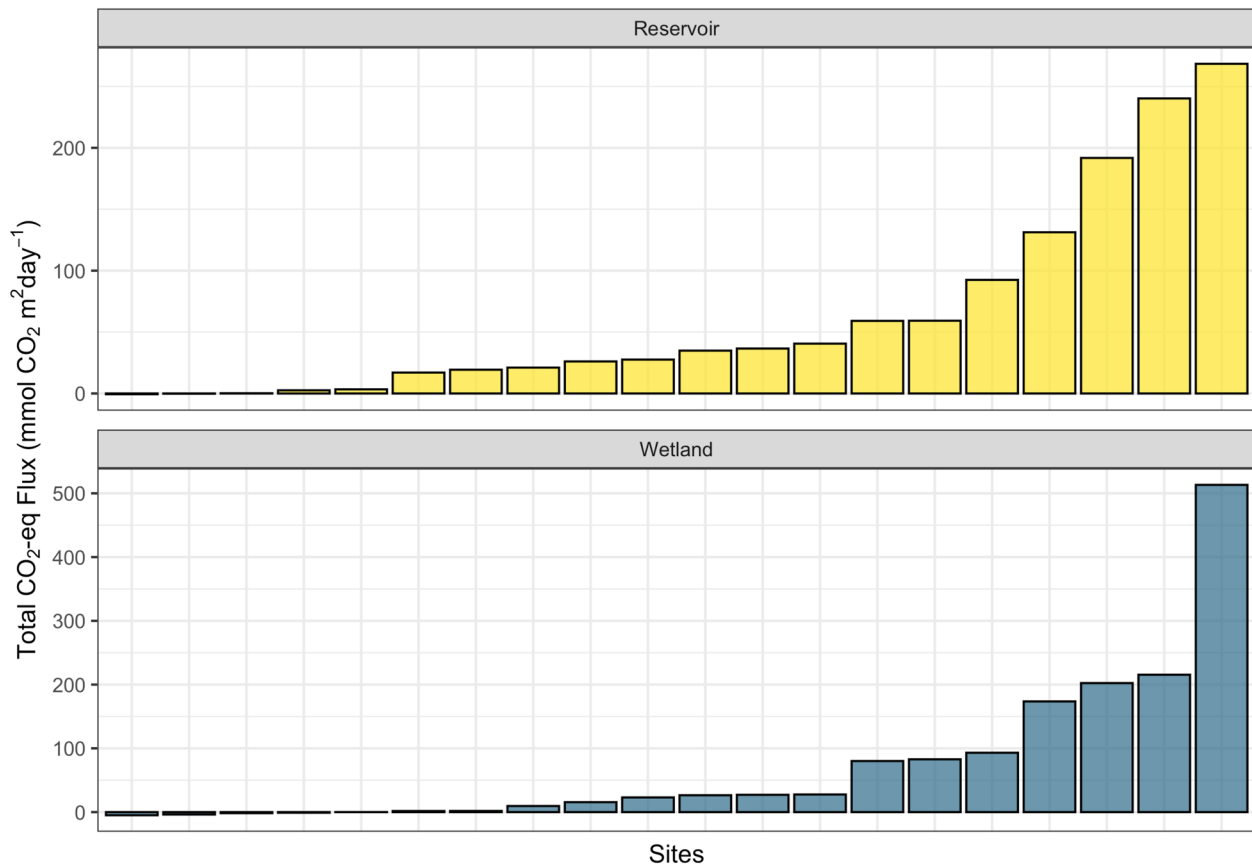
**Figure 7.** Total carbon dioxide ($CO_2$)-equivalent (eq) flux from each reservoir (top) and wetland (bottom). All values include diffusive flux from carbon dioxide, methane, and nitrous oxide. Carbon dioxide equivalence calculated using the 100-year sustained global warming and cooling potentials (Neubauer & Megonigal, 2015). Each bar represents one site. One reservoir and three wetlands are net $CO_2$ sinks. Note difference in $y$-axis scale.

assumption. Similarly, although we did not observe significant differences in wind speed at time of sampling between waterbody type (Table 1), these differences may become apparent over 24-hr periods, and the application of the $k$ values observed in agricultural reservoirs may underestimate $k$ in wetland ponds. Future research should evaluate differences in $k$ values, wind speed, and mixing in different water body types in order to derive more confidence in GHG flux rates.

In the context of up-scaling of carbon budgets for inland waters (Cole et al., 2007; Tranvik et al., 2009), our results present a novel finding that $CO_2$ and $CH_4$ fluxes are comparable between natural wetland ponds and constructed agricultural reservoirs, whereas $N_2O$ fluxes are significantly lower in natural wetland ponds. This result, combined with the finding that the mechanisms controlling the production and consumption of GHGs appear to differ between these waterbody types, suggests that the water body types may respond individualistically to future environmental change and that landscape models of integrated GHG flux should distinguish between natural and artificial waterbodies. We also conclude that natural and artificial waterbodies of similar sizes may exhibit substantially different regulatory mechanisms and that artificial waterbodies should have their own recognition in emission factor models, as suggested for $N_2O$ by Webb et al. (2021). Given that ponds and reservoirs were co-located in similar landscapes, we infer that the overarching differences in GHG dynamics between waterbody types may be driven mainly by differences in morphology, stratification and age of the two basins, factors which affect pond mixing, nutrient content, and ecosystem productivity.

## Conflict of Interest

The authors declare no conflicts of interest relevant to this study.

## Data Availability Statement

All R code is available online in a GitHub repository (https://github.com/finlay4k/Jensen_wetland_dugout.git) and data are available on Zenodo (https://doi.org/10.5281/zenodo.7603722).

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